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Degradation in urban air quality from construction activity and increased traffic arising from a road widening scheme

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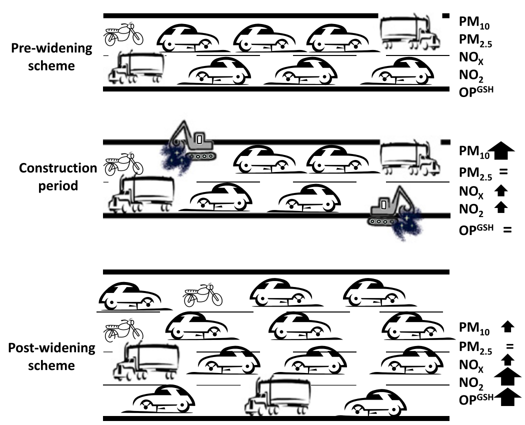
HIGHLIGHTS

- Local air quality deteriorated after completion of a road widening scheme in south London.
- The EU PM_{10} limit value (LV) was breached during construction.
- NO_2 LV was breached after scheme due to increased cars, taxis and LGVs.
- Increase of pro-oxidant components in the PM coarse mode after the road widening.
- Mean PM_{10} emission factor for the construction phase was 0.0022 kg m^{-2} $month^{-1}$.

GRAPHICAL ABSTRACT



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ABSTRACT

Road widening schemes in urban areas are often proposed as a solution to traffic congestion and as a means of stimulating economic growth. There is however clear evidence that new or expanded roads rapidly fill with either displaced or induced traffic, offsetting any short-term gains in eased traffic flows. What has not been addressed in any great detail is the impact of such schemes on air quality, with modelled impact predictions seldom validated by measurements after the expansion of road capacity. In this study we made use of a road widening project in London to investigate the impact on ambient air quality (particulate matter, NO_X, NO_2) during and after the completion of the road works.

PM_{10} increased during the construction period up to 15 μg m^{-3} during working hours compared to concentrations before the road works. A box modelling approach was used to determine a median emission factor of 0.0022 kg PM_{10} m^{-2} $month^{-1}$, three times larger than that used in the UK emission inventory (0.0007 kg PM_{10} m^{-2} $month^{-1}$). Peaks of activity released 0.0130 kg PM_{10} m^{-2} $month^{-1}$, three and eight times smaller than the peak values used in the European and US inventories. After the completion of the widening there was an increase in all pollutants from the road during rush hour: 2–4 μg m^{-3} for PM_{10}; 1 μg m^{-3} for $PM_{2.5}$; 40 and 8 μg m^{-3} for NO_X and NO_2, respectively. NO_2 EU Limit Value was breached after the road development illustrating a notable deterioration in residential air quality. Additionally, PM_{10}, but not $PM_{2.5}$, glutathione dependent oxidative potential increased after the road was widened consistent with an increase in pro-oxidant components in the coarse particle mode, related to vehicle abrasion

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processes. These increased air pollution indices were associated with an increase in the number of cars, taxis and LGVs.

1. Introduction

A large number of studies have shown excess health risks from living in close proximity to roads; however their attribution to single air pollutant is less clear. The adverse health effects from road proximity might be due to tailpipe emissions of airborne particles, particles from non-exhaust sources such as tyre and brake wear or gaseous pollutants including NO_2 (WHO, World Health Organization, 2013).

New or widened roads are often proposed to relieve congestion or to support economic growth, however there is a little appraisal of road schemes once they are built (Matson et al., 2006). It has been shown through case studies in the UK and the Netherlands that the benefits from reduced congestion and shorter journey times are often short lived as new road network capacity is taken up by induced traffic growth (Matson et al., 2006). Few studies have considered the air quality impacts of new road construction and its subsequent operation. The available studies are limited to the construction of urban road tunnels (Bartonova et al., 1999; Cowie et al., 2012) and are thus not directly applicable to most urban road schemes; and a scheme in Antwerp designed to reduce road capacity rather than increase it (Stranger et al., 2008).

There is ample evidence that construction activities are an important source of particulate matter (PM) into the atmosphere and can have a substantial temporary impact on air quality. Construction activities represented 3.8% of total particulate emissions from open sources in the US in 1976 (Evans and Cooper, 1980). Dust and other air pollution from demolition and construction can impact greatly on the health and quality of life of people working on and living close nearby with some studies reporting an increment of mortality due to chronic obstructive pulmonary disease among construction workers (Bergdahl et al., 2004). Emissions of PM during the construction of a building or road are associated with land clearing, ground excavation, cut and fill operations and the construction of a particular facility itself. PM emissions from construction are largely in the coarse fraction but they are also a source of airborne ultrafine particles (Kumar et al., 2012). Dust emissions often vary substantially from day to day depending on the level of activity, the specific operations and the prevailing meteorological conditions making it difficult to assess the total contribution of such emissions to the air pollution levels of a city or region (Chang et al., 1999).

Fugitive emissions from construction are generally poorly quantified in global and national emission estimates as most countries do not report fugitive emissions from construction; and the emission factors (EF) for construction activities are uncertain (Janssens-Maenhout et al., 2012). The Environmental Protection Agency (EPA) in the United States (US) and the Coordinated European Particulate Matter Emission Inventory Program (CEPMEIP) in Europe provide EF for construction activities to generate national primary PM emissions. These EFs express the amount of PM_{10} emitted by area disturbed per month of activity and therefore the quantity of PM produced is not dependent on the type of construction but merely on the area of land disturbed. The EPA also lists operation-specific emission factors for use when detailed information (material, silt content, vehicle weight, speed, etc.) is available (EPA, 1995; 2011).

According to the London Atmospheric Emissions Inventory (LAEI), construction and demolition activities are estimated to account for 1.4% of the total PM_{10} emitted in London in 2010 (LAEI, 2013). Examination of PM_{10} measurements across London suggested that fugitive emissions from construction works were responsible for daily mean concentrations above 50 $\mu g\ m^{-3}$ at 25% of the monitoring sites for each year during 1999–2001 (Fuller and Green, 2004). Construction activities around King's Cross in London were responsible for the excess levels of PM_{coarse} (particulate matter in the size range 10 μm–2.5 μm) in 2003–2004 (Haynes and Savage, 2007).

This study aimed to quantify the impact on the pollution levels for PM_{10}, $PM_{2.5}$, NO_X and NO_2 arising from extensive road works to redevelop the A206, Thames Road, in south-east London, as a dual carriageway. The road improvements involved a 1.8 km section of the A206 Thames Road and were designed to reduce delays on the route and regenerate the area, the second-largest industrial area in London (Bexley, 2002). The road is located close to the river Thames which is reflected in the gravels and alluvium soil types in the area. It was estimated that around 25,000 m^3 of material would be brought in to construct the road and around 18,000 m^3 would be excavated.

The pre-scheme environmental impact assessment included both traffic and dispersion modelling (EPA CAL3QHC) for the widened road. It predicted an increase of the traffic flow of 19% in the morning peak and 9% in the evening peak but only “marginal” changes in local air quality. Under typical meteorological conditions changes in annual mean concentrations were expected to be -0.2 to $+0.1$ $\mu g\ m^{-3}$ for PM_{10} and -0.8 to $+1.8$ $\mu g\ m^{-3}$ for NO_2 depending on the location (Babtie-Bexley, 2001). The air pollution impact of the construction phase was not considered in the pre-scheme impact assessment.

The construction works lasted for one year and seven months, between January 2006 and August 2007. To quantify the increment of air pollutant concentrations from the road widening scheme a unique pairing of air quality monitoring sites was placed in each side of the road in the centre of the construction area. A source apportionment method was applied to distinguish PM_{10} emitted by road traffic from that emitted by the construction activity and a dedicated fugitive construction PM_{10} EF was calculated and compared with those commonly used in emission inventories.

Air pollution for the time periods before and after the redevelopment works was also analysed to assess any air pollution changes arising from the widened road.

2. Material and methods

2.1. Site location and instrumentation

The road works in the study area lasted for one year and seven months, from January 2006 to August 2007. In order to evaluate the impact on the air quality before, during and after the widening works, two periods of one year and seven months before and after the road works are also analysed. The period “before” was taken from May 2004 until December 2005; the “after” period ran from September 2007 to April 2008.

Air quality monitoring station (AQMS) were located on the north (51.4572°N, 0.1953°E) (AQMS-N) and south sides (51.4569°N, 0.1946°E) (AQMS-S) of the A206 Thames Road, in south-east London, separated by a distance of 58 m. The layout of the road runs from north-west to south-east. Two lateral access roads run from north-east (NE) to south-west (SW) close to AQMS-N; and from W to E close to AQMS-S. During the construction period the access road next to the AQMS-N was unpaved. A mix of one and two storey industrial premises was located east of AQMS-N and a two storey residential housing area was located west of AQMS-S. The suburban background concentration was taken from the measurements at Bexley — Slade Green (51.466°N; 0.1848°E) (AQMS-back) located 1.2 km north-west of AQMS-S and AQMS-N.

All AQMSs were equipped with a Tapered Element Oscillating Micro Balance (TEOM) PM_{10} and $PM_{2.5}$ to monitor particulate mass concentrations. Nitrogen oxides ($NO_X = NO + NO_2$) were measured by chemiluminiscence by API 200E analyzers. Fortnightly calibrations enabled the traceability of measurements to National Meteorological Standards. NO_X and PM instruments were subject to twice yearly independent audits. PM_{10} measurements were also converted into PM_{10} equivalent measurements by the Volatile Correction Model (VCM) method (Green et al., 2009). A collocated TEOM-FDMS instrument was installed in AQMS-N. PM_{10} from TEOM VCM correlated well with PM_{10} TEOM-FDMS with correlation coefficients $R > 0.9$ for all three time periods. AQMS-N was also equipped with a meteorological station to measure wind speed and direction, temperature and relative humidity.

Annual mean traffic counts between 7.00 and 18.00 h for the section of A206 Thames Road next to AQMS-N and AQMS-S were provided by the Department for Transport (DfT) and Transport for London (TfL). Traffic data was available for 2005 (representative for the "before" conditions), 2007 ("during") and 2009 ("after"). The data set disaggregated traffic into pedal cycles, motorcycles, cars and taxis, buses, Light Goods Vehicles (LGV), and Heavy Goods Vehicles (HGV) counts. Additional traffic data was also available for other two adjacent sections of the main road network as shown in Fig. 1.

2.2. Road increment and source apportionment

The increment in air pollutant concentrations due to emissions from the road (including tail-pipe, break and tyre wear resuspension; and pollution arising from construction) was calculated as:

$$C_r = C - C_u \quad (1)$$

where C_r is the increment in concentration from the road; C is the concentration measured downwind the road and C_u is the upwind concentration. Here, the concentration measured at either AQMS-N or AQMS-S was considered upwind depending on the wind conditions. Bivariate polar plots calculated by means of the Openair software (Carslaw and Ropkins, 2012) of the mean hourly differences measured in AQMS-N minus AQMS-S were used to identify the wind conditions when each AQMS was upwind. The AQMS-S acted as background for winds blowing from SSE to NW; the AQMS-N acted as background for winds blowing from 0 to 30° north at wind speeds lower than 5 m s^{-1} (Supplementary Fig. 1). Next to the AQMS-N (16 m) there was an unpaved access road running from NE to SE. Large PM_{10} concentrations were measured at high wind speeds when the wind was blowing from the unpaved road and AQMS-N could not be used to calculate the road increment at these times (see Fig. S1). A total of 1560 hourly road PM_{10} increments were negative over a total of 21,452 hourly measurements when road increments could be calculated (7.3% of the data). Further, observations in which the road increment for any of the other pollutants (NO_X, NO_2 or $PM_{2.5}$) was negative were also excluded from calculations. These negative C_r values might be explained by the fact that one site was affected by a very local source (perhaps a car idling next to one of the AQMSs) that was not affecting the downwind site. The total data excluded represented 26.4%.

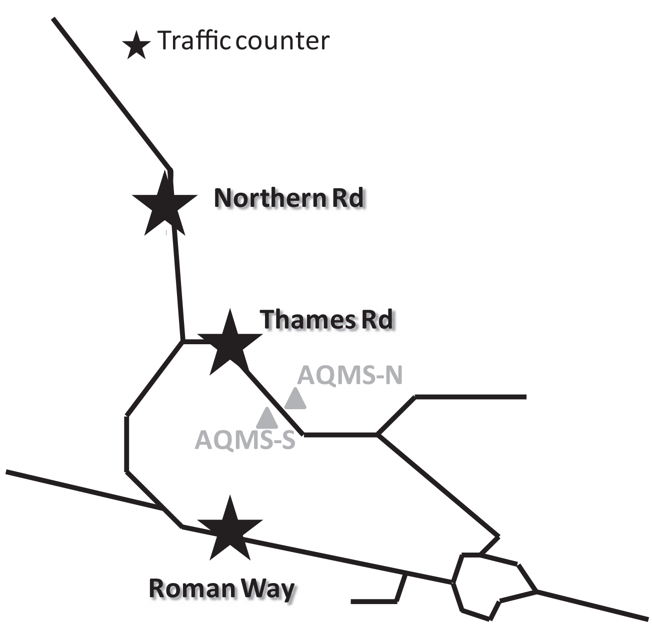

Fig. 1. Schematic map of the main roads around the study area.

NO_X was used as a tracer to quantify the fraction of PM due to traffic emissions and separate this from PM due to construction works (APEG, 1999; Fuller and Green, 2006). This source apportionment distinguishes PM associated with NO_X (PM–NO_X related) which includes stationary sources and tail-pipe and non-tail-pipe road emissions; and PM not associated with NO_X (PM–non-NO_X) which includes secondary aerosols from long-range transport, construction works and natural sources such as marine aerosol and wind-blown resuspension of dust. Long-range transport and natural sources affecting the regional scale (such as marine aerosol) would be included in C_u and C would therefore not affect C_r. For this study, the C_r PM–non-NO_X could therefore be attributed to fugitive emissions arising from construction activities.

The source apportionment emission ratio of NO_X to primary traffic PM was derived from linear relationships of the form $Y = A * X + B$, where X is the explanatory variable (C_r NO_X), Y is the dependent variable and A and B are the gradient and intercept (C_r PM), respectively. Reduced major axis (RMA) regression, also termed standardised major axis, was used due to uncertainty present in the measurements of both dependent and determinant variables (Ayres, 2001; Warton et al., 2006). The PM–NO_X traffic was calculated as:

$$[C_r\text{PM–NO}_X\text{-related}] = A * [C_r NO_X] \quad (2)$$

Hourly C_r PM–NO_X related were calculated using Eq. (2). Fugitive C_r PM–non-NO_X concentrations were then calculated as:

$$[C_r\text{PM–non-NO}_X] = [C_r\text{PM}]–[C_r\text{PM–NO}_X\text{ related}] \quad (3)$$

Data from the periods before and after the construction were used to calculate a representative PM_{10}/NO_X ratio for traffic emissions and labelled 'source apportionment'.

2.3. Real-world PM_{10} emission factor from the road work activity

The emission factor (EF) for PM_{10} from the construction activity was calculated using a simple box model (Jamriska and Morawska, 2001). This is based on a mass balance approach that considers the road as a box with dimensions W (width), L (length) and H (height) with one of the sides parallel with wind direction (Fig. 2). The model assumes i. that the main processes affecting the pollutant concentration (emissions and losses) are in equilibrium; ii. perfect turbulent mixing and that the concentration of pollutants is uniform in the whole volume of air; iii. losses by chemical reactions or gravitational deposition are negligible; iv. no pollutants leave through the top and longitudinal sides of the box.

The mass balance for any pollutant can be written as:

$$F_{lateral\ in} + F_{road} = F_{lateral\ out} \quad (4)$$

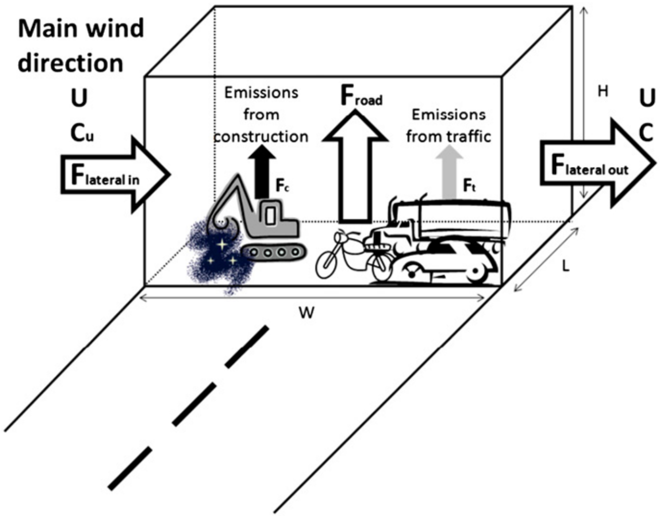

Fig. 2. Schematic diagram of the box model used to calculate PM_{10} emissions from the road construction phase of the road widening.

where $F_{laterial\ in}$ and $F_{lateral\ out}$ are the particulate mass flow rate (in $\mu g\ s^{-1}$) entering and leaving the box, respectively; and F_{road} is the particulate mass flow rate ($\mu g\ s^{-1}$) due to emissions in the road. Those terms are quantified as:

$$F_{lateral\ in} = U \cdot H \cdot L \cdot C_u \tag{5}$$

$$F_{road} = EF \cdot W \cdot L \tag{6}$$

$$F_{lateral\ out} = U \cdot H \cdot L \cdot C \tag{7}$$

where U is the normal component of the horizontal wind velocity expressed in m s^{-1}; W, L and H are the width, length and height of the box, respectively, expressed in m; C_u and C are the upwind and downwind concentrations, respectively, in $\mu g\ m^{-3}$; and EF is the emission rate or emission factor expressed in $\mu g\ m^{-2}\ s^{-1}$.

Substituting terms in Eq. (4) for Eqs. (5)–(7) and arranging parameters, EF is isolated and calculated as:

$$EF = \frac{U \cdot H \cdot (C - C_u)}{W} \tag{8}$$

where C–C_u equals the road increment, C_r, as expressed in Eq. (1).

Calculation of PM_{10} EF from construction activities is carried out as follows:

$$F_{road} = F_{traffic} + F_{construction} \tag{9}$$

$$W \cdot L \cdot EF_{road} = W \cdot L \cdot EF_{traffic} + W \cdot L \cdot EF_{construction} \tag{10}$$

$$EF_{road} = \frac{U \cdot H \cdot C_r PM_{10}}{W} \tag{11}$$

$$EF_{traffic} = \frac{U \cdot H \cdot (A \cdot C_r \cdot NO_x)}{W} \tag{12}$$

where A is the gradient from source apportionment in Eq. (2).

The normal component of the mean vertical velocity (U) is calculated from the mean horizontal wind speed (WS) and wind direction (WD) measured in AQMS-N as:

$$U = WS \cdot \cos(WD - \alpha) \tag{13}$$

where α is the angle between the road direction and the north (here, 49°).

The height of the box H is the maximum altitude where pollutants mix. Here, measurements taken at different heights were not available, so, the value of 8.4 m was assumed on the basis of measurements undertaken by Jamriska and Morawska (2001).

2.4. Oxidative potential

PM_{10} and $PM_{2.5}$ samples were collected on TEOM filters from the AQMS-S to assess changes in oxidative potential before, during and after the road widening. Particles were extracted from the filters and PM oxidative potential assessed with OP calculated based on the extracts' capacity to deplete ascorbate and glutathione from a simple synthetic respiratory tract lining fluid model over four hour incubation at 37 °C, pH7.0 (Godri et al., 2010; Kelly et al., 2011). OP data were expressed as glutathione and ascorbate dependent oxidative potentials (OP^{GSH} and OP^{AA}), expressed per μg of extracted PM and per m^3 of ambient air. A total oxidative potential (OP^{TOT}) was also calculated based on the sum of the two previous measures (Godri et al., 2010).

3. Results

3.1. Pollution concentrations and sources

The European Union Limit Values (LV) stipulates that annual PM_{10} should not exceed 40 $\mu g\ m^{-3}$ and the number of days with daily PM_{10} concentrations greater than 50 $\mu g\ m^{-3}$ should not exceed 35. The LV for NO_2 is 40 $\mu g\ m^{-3}$ as annual mean. These statistics were calculated for AQMS-N, AQMS-S and at the suburban background (AQMS-back) for 2004 and 2005 (as representative of the conditions before the construction works); for 2006 and 2007 (as representative for the conditions during the road works); and 2008 and 2009 (as representative for the conditions after the road works). Results are summarized in Table 1. Increments above the background concentrations (roadside-background), labelled as local increments, are also summarized. Annual PM_{10} was below the 40 $\mu g\ m^{-3}$ LV for all years. However, the number of days with daily mean $PM_{10} > 50\ \mu g\ m^{-3}$ increased at both roadside AQMS and breached the limit of 35 days in both years affected by the construction (2006 and 2007). Both AQMS achieved the NO_2 annual LV before the construction (except AQMS-S in 2005) but failed afterwards, in 2008 and 2009 at AQMS-N; and in 2008 in AQMS-S. AQMS-N also breached the NO_2 LV during the construction period.

The redevelopment of the Thames Road to a dual carriageway added 3 $\mu g\ m^{-3}$ to local PM_{10}; 6.5 $\mu g\ m^{-3}$ to local NO_2 at AQMS-N and 3 $\mu g\ m^{-3}$ to local NO_2 at AQMS-S on annual basis (values in brackets in Table 1). These changes represented 7.5% and 8–16% of the annual mean threshold concentration for PM_{10} and NO_2, respectively. AQMS-N was downwind the road most of the times and therefore measured higher local increments than AQMS-S (see wind rose plot in Supplementary Fig. S2).

The traffic data showed an increase in the total vehicle flow on Thames Road (Fig. 3a) after the widening works. This increase was mainly due to an increase in the number of cars and taxis (Fig. 3b) at all hours of the day (between 11 and 19%) but most notably during the rush hour peaks (32–49%) and to lesser extent to LGVs (Fig. 3c) which increased by 50% at 6 am and 12–18% between 2 and 4 pm. Surprisingly, the number of HGVs did not increase after completion of the road works; these were lower than "before" conditions between 9 am and 12 pm and between 5 and 8 pm (Fig. 3d). However HGV counts

Table 1
Annual PM_{10} and NO_2 concentrations with the number of daily PM_{10} concentrations were greater than 50 μg m^{-3} for the suburban background AQMS (back), and the two AQMS along the A206 Thames Road (AQMS-N and AQMS-S) for 2004 and 2005 ("before" conditions), 2006 and 2007 ("during"); and 2008 and 2009 (for the "after" period). Values in brackets are the increment above the suburban background.

	Site	Before		During		After	
		2004	2005	2006	2007	2008	2009
Annual mean PM_{10} (μg m^{-3})	Back	23	23[a]	23	23	18	18
	Road-N	29 (6)	28 (5)[b]	35 (12)	30 (7)	28 (10)	25 (7)
	Road-S	25 (2)	26 (3)	30 (7)	28 (5)	24 (6)	23 (5)
No. days daily PM_{10} > 50 μg m^{-3}	Back	7	12	8	23	5	2
	Road-N	27 (20)	23 (11)[b]	59 (51)	46 (23)	32 (27)	16 (14)
	Road-S	18 (11)	26 (14)	49 (41)	36 (13)	31 (26)	12 (10)
Annual mean NO_2 (μg m^{-3})	Back	34	36	36	34	33	32
	Road-N	40 (6)	40 (4)[c]	44 (8)	41 (7)	45 (12)	43 (11)
	Road-S	38 (4)	41 (5)	38 (2)	37 (3)	41 (8)	39 (7)

[a] Data capture rate less than 90% for the year (70%).
[b] Data capture rate less than 90% for the year (81%).
[c] Data capture rate less than 90% for the year (84%).

did increase between 6 and 11 am during the construction period, possibly due to construction related traffic.

Table 2 shows the distribution of hourly road increments (C_r) and daily mean road increments for each time period of the study, calculated using Eq. (1). The 5th percentile and median values of hourly C_r PM_{10} averages did not change during the road works. However mean C_r PM_{10} increased by 4 μg m^{-3} and 95th percentile increased by 3.9 μg m^{-3} during the construction period, consistent with previous observations of construction giving rise to short-term high level concentrations (Fuller and Green, 2004). Despite an increase in PM_{10} during rush hour peaks after the completion of the road widening, the mean daily PM_{10} concentration did not change. No changes were observed in the distribution of hourly mean C_r $PM_{2.5}$ averages for any of the periods. During the construction period an increase of 3.5 ppbv of NO_X was observed on average followed by a larger increase of 9.9 and 12.5 ppbv respectively in the mean and 95th percentile after the completion of the road works. NO_2 increased

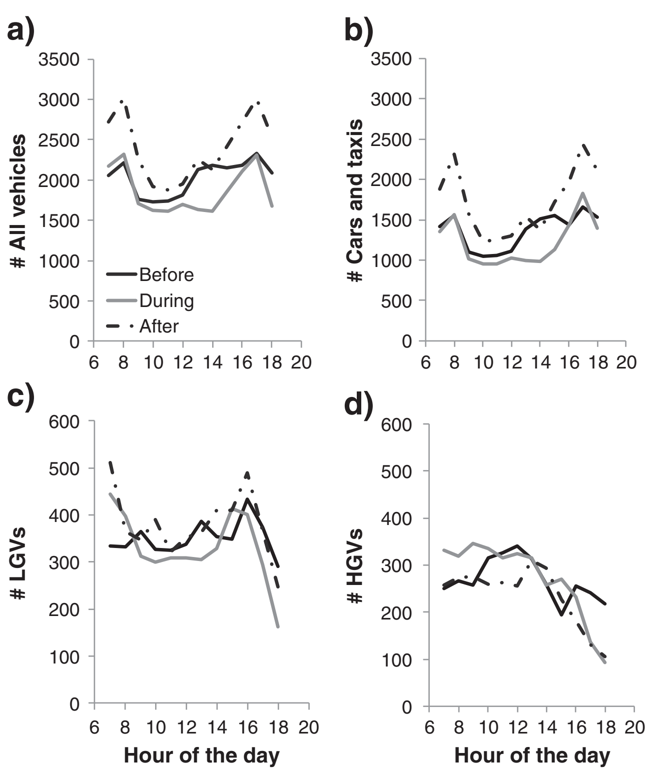

Fig. 3. Annual mean traffic counts for before, during and after the road works along the A206 Thames Road: a) all vehicles; b) cars and taxis; c) LGVs and d) HGVs.

during and after the construction period. During the road works an increase of 2.1 and 2.6 ppbv was observed in the mean and 95th percentile, respectively; after the road works, the increase was 2.8 and 3.9 ppbv for the same statistics.

Mean C_r PM_{10} from the road was higher (up to 15 μg m^{-3}) from 6 am to 5 pm during the construction period (Fig. 4a) when compared with periods outside the construction. The mean C_r $PM_{2.5}$ daily cycle did not experience major changes on its pattern; however, an increment of ~1 μg m^{-3} was observed during the morning and afternoon rush hour peaks after the construction (Fig. 4b), consistent with the increase in cars and taxis at this time; and to a lesser extent, with the increase in LGVs. Comparing the diurnal cycle for C_r NO_X increased by ~5 ppbv during the construction and by a further ~15 ppbv after construction, with increments during the morning rush hours peaking ~25 ppbv with the completed road layout. C_r NO_2 increased by ~2 ppbv during and after construction at all times. After the construction, C_r NO_2 concentrations in the morning rush hour increased by ~4 ppbv compared to the period before the works. The mean diurnal cycle for $PM_{2.5}$ was similar for all three periods of time studied, therefore the enhancements of PM_{10} during the road works were due to coarse PM (10 μm–2.5 μm); another indication that road increments in PM_{10} were due to fugitive rather than exhaust emissions.

The mean C_r concentrations were higher during weekdays (mean diurnal average of 5.5 μg m^{-3} for PM_{10}, 2.5 μg m^{-3} for $PM_{2.5}$; 35.4 ppbv for NO_X and 9.0 ppbv for NO_2) than weekends (2.8 μg m^{-3} for PM_{10};

Table 2
Distribution of hourly C_r averages (5th percentile, median, mean and 95th percentile) and daily mean concentration for all pollutants for the three periods of time considered.

		Before	During	After
C_r PM_{10} (μg m^{-3})	Hourly 5th p	2.4	2.3	2.0
	Hourly median	4.4	4.9	4.4
	Hourly mean	6.4	10.4	6.3
	Hourly 95th p	7.7	10.6	8.3
	Daily mean	6.2	11.0	6.2
C_r $PM_{2.5}$ (μg m^{-3})	Hourly 5th p	1.2	0.9	1.1
	Hourly median	2.2	1.9	2.3
	Hourly mean	2.7	2.6	2.9
	Hourly 95th p	3.7	3.5	3.9
	Daily mean	2.7	2.9	2.9
C_r NO_X (ppbv)	Hourly 5th p	12.5	15.3	17.6
	Hourly median	25.0	28.3	33.5
	Hourly mean	32.1	35.6	43.0
	Hourly 95th p	43.9	48.2	56.4
	Daily mean	30.3	37.1	40.5
C_r NO_2 (ppbv)	Hourly 5th p	2.7	4.5	4.5
	Hourly median	5.9	8.0	8.6
	Hourly mean	7.1	9.2	9.9
	Hourly 95th p	10.0	12.6	13.9
	Daily mean	7.6	10.4	10.6

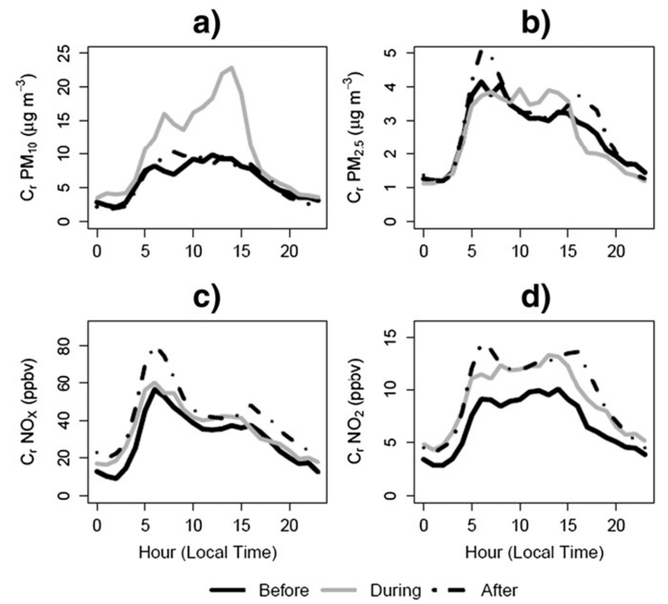

Fig. 4. Mean diurnal cycle for the road increment (C_r) for PM_{10} (a), $PM_{2.5}$ (b), NO_X (c) and NO_2 (d).

1.3 µg m^{-3} for $PM_{2.5}$; 15.7 ppbv for NO_X and 5.0 ppbv for NO_2) consistent with lower traffic flows during weekends. C_r PM_{10} concentrations were higher on Saturdays from 6 am to 12 pm during the road work period (Supplementary Fig. 3). That is an indication that the PM_{10} enhancements during those hours arose from construction works and not from traffic emissions that were expected to be lower on Saturdays.

Mean C_r PM_{10} and C_r NO_X for each hour of the week were calculated and showed positive correlation ($R^2 > 0.65$) for the periods "before" and "after" indicating that both PM_{10} and NO_X had a common traffic source (Fig. 5). The degree of correlation of C_r PM_{10} against C_r NO_X was lower during the construction works ($R^2 = 0.57$) (Table 3). The PM_{10}/NO_X ratio (0.459 ± 0.046 µg m^{-3} ppbv^{-1}, as mean and two standard deviations) doubled compared with the periods "before" (0.194 ± 0.016 µg m^{-3}ppbv^{-1}) and "after" (0.170 ± 0.016 µg m^{-3} ppbv^{-1}). The increase in the PM_{10}/NO_X ratio during construction indicates that more PM_{10} was emitted per ppbv of NO_X; emitted consistent with emissions from construction activities being mainly PM_{10} and not NO_X. C_r $PM_{2.5}$ was strongly positively correlated ($R^2 > 0.80$) with C_r NO_X and $PM_{2.5}/NO_X$ slopes were similar (within the uncertainty limits) for all periods of time (Table 3). This indicates that the main source of $PM_{2.5}$ from the road was traffic related and that the construction did not emit measureable $PM_{2.5}$. As shown in Fig. 5 the regression intercepts were around zero for PM_{10} before and after construction; and around zero for all periods for $PM_{2.5}$; consistent with the assumption that all PM from the road was due to traffic outside the construction periods.

Table 3
Mean slope calculated from linear regressions of PM against NO_X (±2σ), coefficients of determination (R^2) and number of observations (N) for the linear regressions.

	PM_{10}		$PM_{2.5}$		N
	PM_{10}/NO_X (µg m^{-3} ppbv^{-1})	R^2	$PM_{2.5}/NO_X$ (µg m^{-3} ppbv^{-1})	R^2	
Before	0.194 ± 0.016	0.69	0.073 ± 0.004	0.85	168
During	0.459 ± 0.046	0.57	0.080 ± 0.006	0.82	168
After	0.170 ± 0.016	0.65	0.069 ± 0.004	0.91	168
Source apportionment	0.170 ± 0.012	0.61	0.067 ± 0.003	0.85	336

3.2. Construction PM_{10} and emission factor

The source apportionment equations (Eqs. (2) and (3)) were applied to calculate hourly C_r PM_{10}–NO_X related and C_r PM_{10}–non-NO_X concentrations for the construction period using the PM_{10}/NO_X ratio summarized in Table 3 as 'source apportionment'.

The behaviour of the fugitive C_r PM_{10}–non-NO_X was considered under different meteorological conditions (wind speed, temperature and relative humidity). As shown in Fig. 6 fugitive C_r PM_{10} concentrations increased with higher wind speeds and greater temperatures. The positive correlation between temperature and the C_r PM_{10}-fugitive concentrations might be reflective of higher temperatures drying out dusty surfaces and enhancing suspension of particles from the clay soils in the construction area. Conversely, fugitive concentrations decreased with greater relative humidity consistent with the hygroscopic nature of coarse particles as pointed by Kassomenos et al. (2012) making it more difficult for suspension of particles at high humidity values.

The median, 5th and 95th percentiles from hourly C_r PM_{10}, C_r NO_X and wind speed were calculated for 6 am to 5 pm on weekdays and 6 am to 12 pm on Saturdays during the construction period to calculate PM_{10} EF as shown in Table 4. The 5th, median and 95th percentile EFs

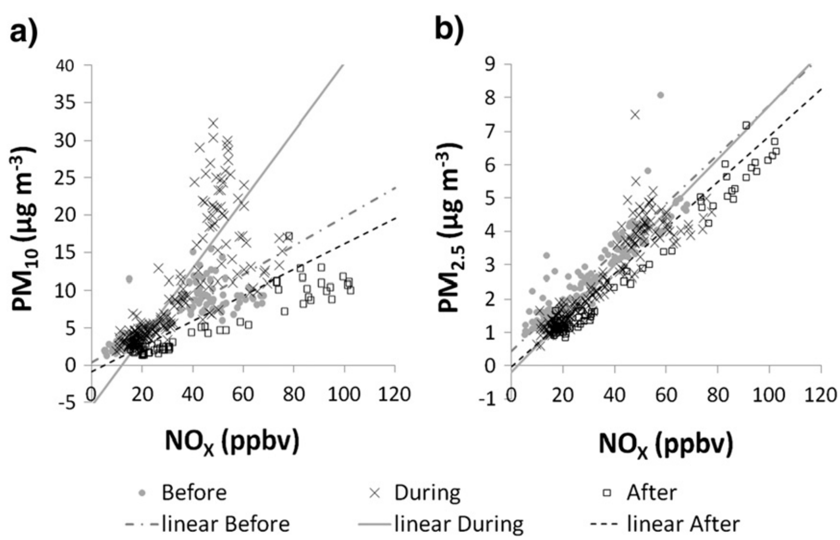

Fig. 5. Relationships between the mean hourly PM_{10} (a) and $PM_{2.5}$ (b) concentrations from the road for each day of the week against mean hourly concentrations of NO_X.

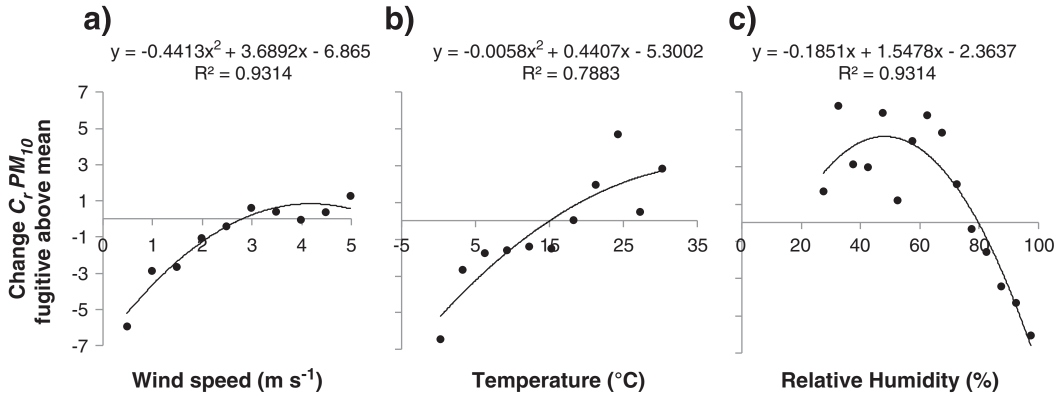

Fig. 6. Median fugitive C_r PM_{10}–non-NO_X for the period of time when road works took place against wind speed (a), temperature (b) and relative humidity (c).

for each sector provides an indication of the temporal variability of emissions during different phases and intensity of the construction activity. The range of EF from all sectors indicated the spatial variability of PM_{10} emissions in different sectors of the road close to the AQMSs. The EF for construction activities, calculated from median values, ranged from 0.1 to 2.2 μg PM_{10} m^{-2} s^{-1}. The largest EFs arose from south of the AQMSs between 190 and 220°N where a new roundabout was constructed. PM_{10} $EF_{construction}$ calculated from 5th percentile values was close to zero or slightly negative (ranging from −0.4 to −0.3 μg PM_{10} m^{-2} s^{-1}) for all wind sectors. That was due to large C_r PM_{10}–NO_X related concentrations leading to slightly negative fugitive C_r PM_{10}–non-NO_X. Peak activity, as calculated from the 95th percentile, led to EF between 20.0 and 130.4 μg PM_{10} m^{-2} s^{-1}.

The median EF for the Thames Road works, on the basis of 61 hours of construction activity per week (11 h at each weekday and 6 h at weekends) from all wind sectors was 0.0022 kg PM_{10} m^{-2} $month^{-1}$. If only C_r PM_{10} and C_r NO_X for non-rainy hours were used, the construction EF was 0.033 kg PM_{10} m^{-2} s^{-1}. This result is in accordance with the effect of relative humidity on C_r PM_{10} (Fig. 6). The worst scenario EF, taken as the maximum monthly EF, was 0.0130 kg PM_{10} m^{-2} $month^{-1}$ (all data) and 0.0141 kg PM_{10} m^{-2} $month^{-1}$ (non-rainy hours) (Table 5).

3.3. Oxidative potential

The glutathione dependent oxidative potential (OP^{GSH}) of PM_{10} at AQMS-S increased significantly after the road widening scheme compared with both the 'before' and construction periods when expressed on a per unit mass basis (Fig. 7). A similar increase in ascorbate dependent OP (OP^{AA}) was also seen between the construction and 'after' period, but for this metric the values after construction were similar to those before the scheme, suggesting that the construction related PM was less oxidative than that derived from traffic. The overall combined metric OP^{TOT} was also found to be increased after the road widening (Table 6). When these data were adjusted to reflect ambient PM oxidative potentials per m^3 only the increase in OP^{GSH}/m^3 remained significant following the construction period (Table 6). No change in $PM_{2.5}$ oxidative potential was noted across the three study periods, either per μg or m^3, implying that the increased OP^{GSH} values reflected enrichment of pro-oxidant components in the coarse mode.

4. Discussion and conclusions

The widening of the A206 Thames Road in London afforded a unique opportunity to address the impact of the construction works on air pollution, as well as its longer term consequences following completion. The construction activity was a source of airborne particles and an increment in the number of daily breaches of the European Union daily mean limit value concentration (50 μg m^{-3}) during the construction phase was observed. PM_{10} concentrations from the road increased up to ~15 μg m^{-3} between 6 am and 5 pm during weekdays and between 6 am and 12 pm on Saturdays in accordance with the timing of construction activities whereas changes in NO_X concentrations were negligible. Most of the particulate matter emitted during construction was in the coarse fraction and its behaviour against different weather parameters was similar as described in other studies. The suspension of coarse particles was enhanced at high wind speeds as previously observed at other urban monitoring sites in Birmingham and Marylebone Road in London (Harrison et al., 2001; Charron and Harrison, 2005). In our study increased relative humidity was linked to a decrease in construction PM and it is therefore possible that wetting the construction site or applying wetting agents such as Calcium–Magnesium Acetate (CMA) might be an efficacious control measure for these sources (Hafner, 2012). The London Best Practice Guidance (GLA-LC, 2006) for construction stipulates a trigger level of 250 μg m^{-3} of PM_{10} (as 15 min mean). During the Thames Road widening the PM_{10} trigger level was breached 158 times at AQMS-N and 81 times at AQMS-S. Had the guidance been available, adherence to the trigger levels might have therefore been

Table 4
5th percentile (5thp), median (50thp) and 95th percentile (95thp) for hourly C_r PM_{10}, C_r NO_X, estimated C_r PM_{10}–NO_X related, measured wind speed and median values of wind direction for weekdays between 6 am and 5 pm and Saturdays between 6 am and 12 pm when the construction works took place at different sectors of the road.

Weekdays 6 am–5 pm Saturdays 6 am–12 pm	0–30° N			160–190° N			190–220° N			220–250° N			250–270° N		
	5th p	50th p	95th p	5th p	50th p	95th p	5th p	50th p	95th p	5th p	50th p	95th p	5th p	50th p	95th p
C_r PM_{10} (μg m^{-3})	14.1	40.8	88.0	25.0	70.1	124.5	22.4	56.1	109.3	16.2	38.3	90.5	11.7	33.5	79.1
C_r NO_X (ppbv)	1.7	8.5	34.9	3.3	20.2	127.7	2.7	14.5	104.3	1.9	8.5	44.3	1.4	7.0	29.1
C_r PM_{10}–NO_X (μg m^{-3})	2.4	6.9	15.0	4.2	11.9	21.2	3.8	9.5	18.6	2.8	6.5	15.4	2.0	5.7	13.4
Wind speed (m s^{-1})	1.0	2.0	2.8	1.1	2.3	3.6	1.0	2.5	3.9	1.2	3.0	4.5	1.2	2.5	4.0
Wind direction (° N)[a]	1.0	16.2	28.7	162.8	177.9	189.2	191.4	205.2	218.9	222.8	237.8	247.9	250.7	259.5	269.0
Total EF (μg m^{-2} s^{-1})	0.6	6.1	35.1	0.9	12.6	123.7	1.0	14.2	158.7	0.9	10.8	83.5	0.6	6.2	42.3
Traffic EF (μg m^{-2} s^{-1})	0.9	5.0	15.0	1.2	7.4	20.5	1.5	9.4	28.3	1.3	8.2	29.0	0.8	5.1	19.5
Construct EF (μg m^{-2} s^{-1})	−0.3	1.1	20.0	−0.3	5.1	103.1	−0.4	4.8	130.4	−0.4	2.5	54.5	−0.3	1.1	22.8

[a] The median value was considered for all cases.

Table 5
Default and worse case conditions of PM_{10} emission factors (EF) from construction activities as calculated for the road works in the Thames Road (this study) and those from emission inventories.

kg PM_{10} m^{-2} $month^{-1}$	Median PM_{10} $EF_{construction}$	Worst conditions of PM_{10} $EF_{construction}$
Thames Road (this study — all data)	0.0022	0.0130
Thames Road (this study — non-rainy hours)	0.0033	0.0141
NAEI (UK)[a]	0.0007	–
CEPMEIP (Europe)[b]	0.0068	0.0448
EPA (US)[c]	0.0272	0.1038

[a] NAEI (National Atmospheric Emissions Inventory) (2013).
[b] EMEP-EEA (2013).
[c] EPA (2011).

effective in controlling PM_{10} during the construction period. The absence of a clear change in the $PM_{2.5}$ and NO_X during the construction suggests that any tailpipe emissions from construction machinery were small compared to the fugitive coarse PM.

The median PM_{10} EF calculated from the road works in the A206 Thames Road (0.0022 kg PM_{10} m^{-2} $month^{-1}$) was three times larger to the one used in the UK National Atmospheric Emissions Inventory (NAEI) (0.0007 kg PM_{10} m^{-2} $month^{-1}$). However, the worst PM_{10} EF from Thames Road was smaller (3 and 8 times smaller) than the ones used in the European and US inventories. This implies that PM_{10} emissions from construction activities might be overestimated for peak activities when modelling from European (EMEP-EEA, 2013) and US inventory EF but underestimated using the current UK default EF.

Importantly our data demonstrate that the transformation of Thames Road into a dual carriage-way road to improve traffic congestion led to a deterioration in residential air quality. This was observed in the increment of air pollutant levels from the road during rush hour peaks: PM_{10} (2–4 μg m^{-3}), $PM_{2.5}$ (1 μg m^{-3}), NO_X (20 ppbv) and NO_2 (4 ppbv) above the levels before the widening scheme. Both AQMS breached the European Union Limit Value for NO_2 after the road works were completed. This increase in pollutant concentrations was mainly attributed to a higher number of cars and taxis and, to a lesser extent, an increase in LGVs in Thames Road. Traffic in the local network did not experience a change indicating that the Thames Road widening scheme did not induce new traffic during its first two years of operation but it might be too early to draw firm conclusions with respect to the effects of the road widening on the longer term operation of the local road network. The new traffic accommodated in Thames Road appeared to be displaced from other areas (Fig. 8).

From an air quality perspective, measurements of air pollutants in other local roads are not available and it is not possible to say if the displacement of traffic onto Thames Road led to an improvement of residential air quality elsewhere. However, the widened road experienced a notable increase in peak hour traffic, more than twice the increase predicted in the pre-scheme assessment. Increases in peak hour traffic have also been observed in other many road expansions since drivers no longer have to adjust their peak route and hour travel times to avoid the most congested routes and periods. Both these effects can negate the projected benefits of road improvements to reduce congestion (DfT,

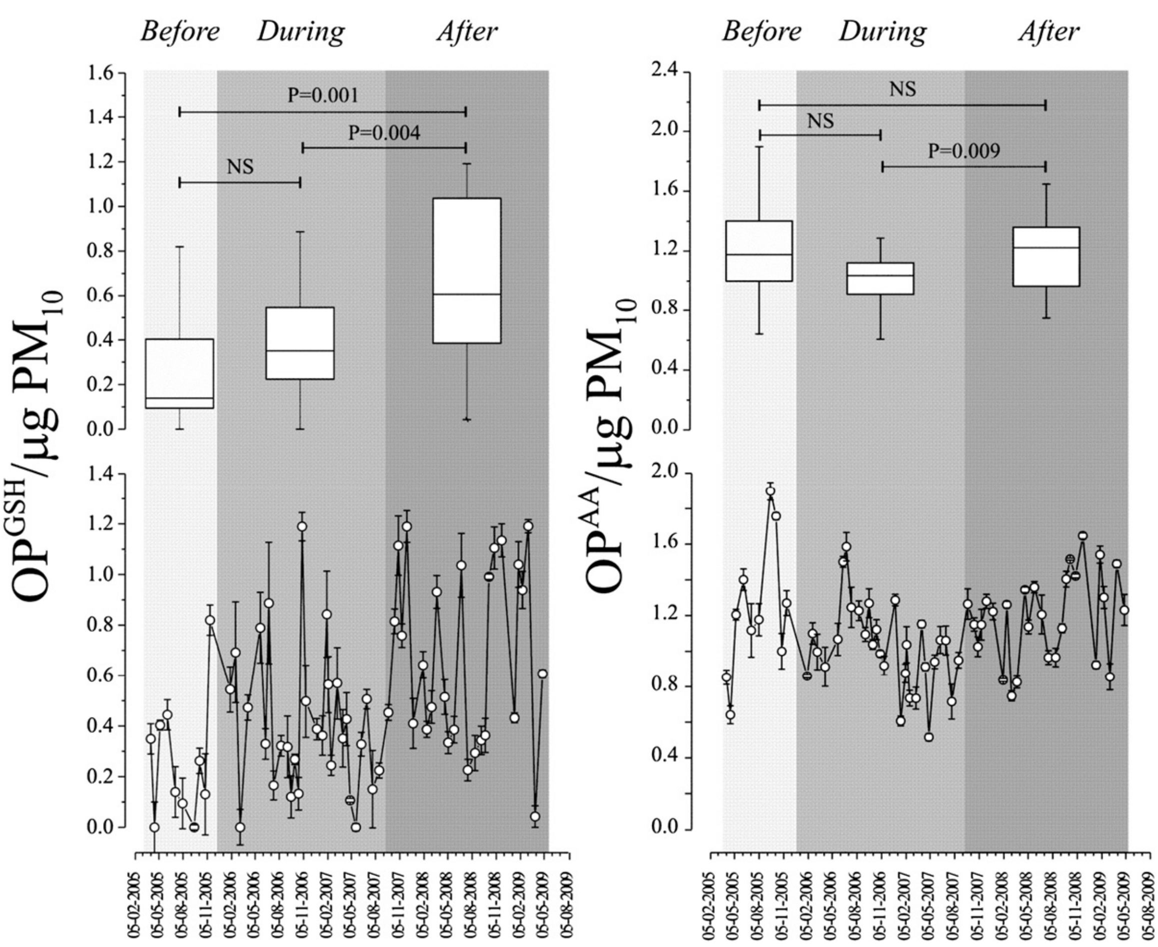

Fig. 7. Glutathione and ascorbate dependent oxidative potential at AQMS-S. The lower panel illustrates the time series of OP data with each value (mean ± 1σ of triplicate determinations) plotted to the mid-point of the filter sampling period. The upper panel illustrate the summary data for each period as a box plot. Statistical comparison of the PM OP values across each sampling interval was performed using the Kruskal–Wallis one-way analysis of variance by ranks, with post-hoc testing between groups using the Mann–Whitney *U* test.

Table 6
Mean (and range) glutathione, ascorbate and total oxidative potentials for $PM_{2.5}$ and PM_{10} sampled for the periods before, during and after the road widening scheme. Oxidative potential is expressed per unit volume of sampled PM (μg) and volume of air (m^3).

		$PM_{2.5}$		PM_{10}	
		μg	m^3	μg	m^3
OP^{GSH}	Before	0.32 (0.17–0.45)	4.5 (2.4–6.3)	0.20 (0.10–0.39)	6.0 (2.2–9.8)
	During	0.32 (0.17–0.52)	4.5 (2.2–5.9)	0.35 (0.22–0.55)	10.1 (5.3–20.5)
	After	0.42 (0.24–0.72)	4.7 (3.0–8.2)	0.61 (0.39–1.01)*≠	16.3 (11.3–23.9)*≠
OP^{AA}	Before	1.25 (1.08–1.36)	15.4 (13.9–21.1)	1.19 (1.03–1.37)	28.8 (25.2–39.2)
	During	1.13 (0.90–1.28)	15.1 (10.9–18.4)	1.04 (0.91–1.12)	29.5 (20.9–35.6)
	After	1.11 (0.91–1.27)	15.0 (10.0–18.7)	1.22 (1.00–1.35)≠	30.5 (23.0–43.1)
OP^{TOT}	Before	1.55 (1.34–1.77)	22.3 (18.8–25.2)	1.44 (1.22–1.88)	36.6 (29.4–46.5)
	During	1.50 (1.27–1.62)	19.4 (14.4–23.3)	1.41 (1.25–1.60)	38.5 (25.8–60.6)
	After	1.55 (1.33–1.71)	20.9 (15.7–23.7)	1.77 (1.51–2.19)≠	48.8 (36.4–60.3)

*$p < 0.05$ values after vs. before the scheme. ≠$p < 0.05$ values after vs. during the road widening. Statistical analysis was as outlined in Fig. 6.

1994; Matson et al., 2006). Industrial regeneration through the road scheme might be judged in terms of increased HGV traffic serving the new and expanded businesses. In this case HGV flows were lower after the scheme was competed compared with the situation before the scheme began. This might also be a function of the economic downturn experienced in the UK, which induced a 10% decrease in national HGV traffic between 2007 and 2009 (DfT, 2013). It is therefore too early to reach a judgement with respect to business regeneration based on data from this study.

The deterioration in air quality following the completion of the works is in stark contrast to the pre-scheme environmental impact assessment. No change in PM_{10} concentrations were predicted at the receptor close to the AQMS-N and NO_2 was predicted to decrease by 0.8 μg m^{-3}. Close to AQMS-S a very slight increase was predicted in PM_{10} (0.1 μg m^{-3}) and NO_2 (0.2 μg m^{-3}) but overall the assessment concluded that changes would be “marginal” and the NO_2 Limit Value would continue to met by under typical meteorological conditions (Babtie-Bexley, 2001). It is likely that the observed increase in NO_2 was in part due to more primary emissions from NO_2 from road traffic in the UK from approximately 1998 onwards (Carslaw, 2005; Carslaw et al., 2011). However, the measured increase in air pollution from the road, and the increased residential concentrations, were not predicted in the environmental impact assessment. Lieplapa and Blumberga (2012) highlighted that the Environmental Impact Assessment (EIA) of 14 projects on motor roads in Latvia between 2001 and 2011 predicted almost no changes on either PM and NO_X concentrations despite higher predicted traffic flows. However, the authors found that PM and NO_X concentrations after the construction of a new road or redevelopment of an existing one increased linearly with the increment of the induced traffic flows. Thus, on the basis of the acquired EIA results it appears that the development of roads does not actually make negative impact on the air quality. We, therefore, highlight the need for routine post-scheme measurements of the impacts of major road projects.

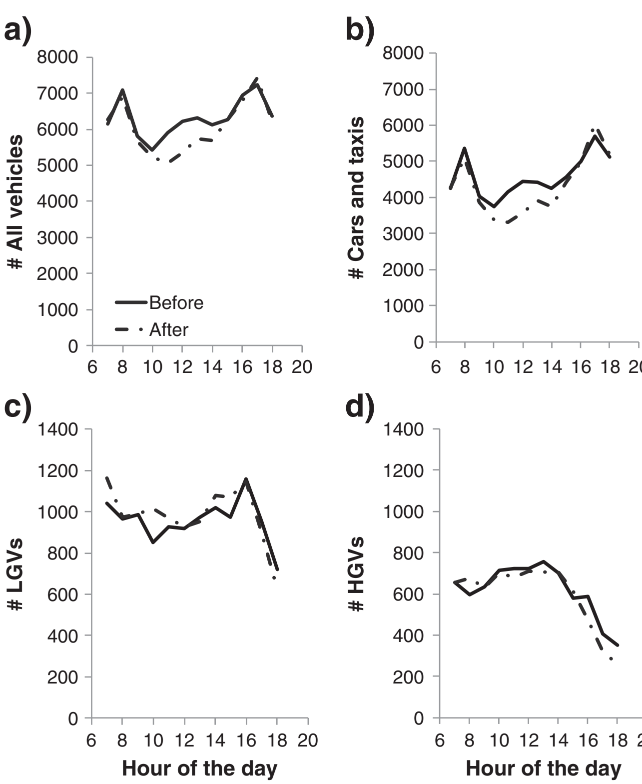

Fig. 8. Total traffic counts for the road network (see Fig. 1); the sum of traffic counts from Northern Road, Thames Road and Roman Way, before and after the road widening scheme: a) all vehicles; b) cars and taxis; c) LGVs and d) HGVs.

The glutathione and ascorbate dependent oxidative potentials of PM_{10} and $PM_{2.5}$ at AQMS-S over the pre- to post-road widening scheme were assessed. Previous work has shown that these two measures are sensitive to different PM constituents and sources, with the glutathione-dependent metric being sensitive to local traffic sources, predominately components related to vehicular abrasion processes and the ascorbate dependent metric being more sensitive to regional PM components (Kelly et al., 2011; Mudway et al., 2011). After the road widening scheme we observed a significant increase in oxidative potential, particularly in the glutathione dependent metric, though this was only apparent in PM_{10} and not $PM_{2.5}$ samples. This was consistent with an increase in pro-oxidant components in the coarse particle mode arising from the road.

Our study was limited to considering the air quality impacts on the widened road itself and was not able to assess impacts that may have occurred on the surrounding road network. It is also likely that our assessment was affected by the economic downturn experienced in the UK at the time of the completion of the scheme. This would have temporarily lessened the impact on residential air quality in 17 months after the widened road was opened.

In conclusion, using the widening of the A206 Thames Road in London as the basis for a detailed assessment of air quality before, during and post-construction, we have obtained data demonstrating the air pollution impacts of road widening scheme and that the expanded road capacity resulted in more local traffic and degraded local residential air quality. Our data also illustrates an important mismatch between modelled impact assessments and the measured reality after the completion of the scheme highlighting the need for post-scheme assessments.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2014.07.060.

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